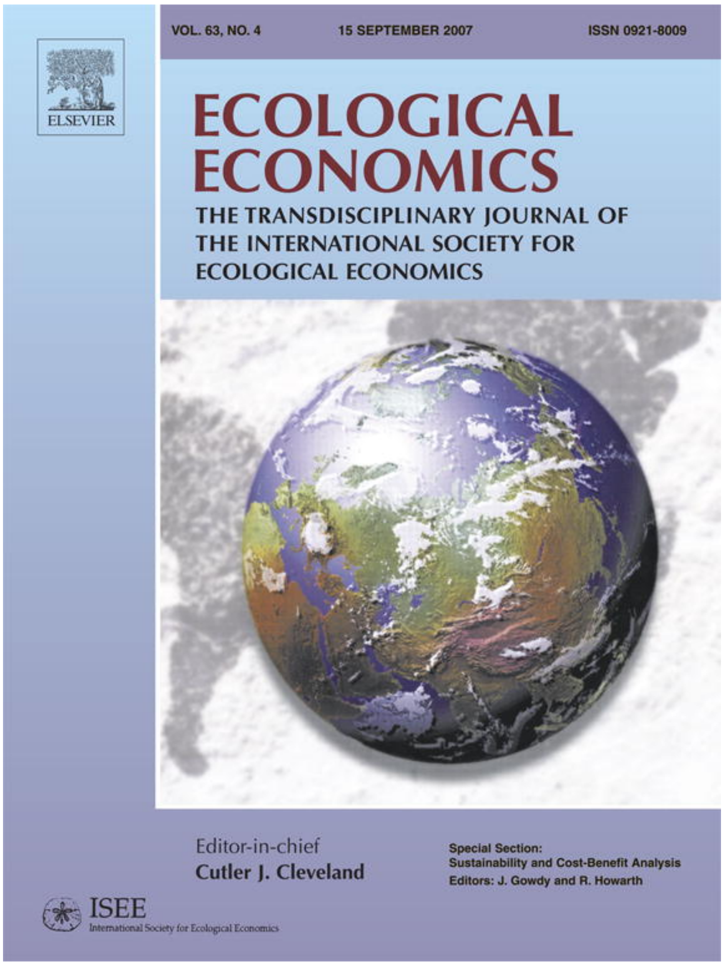
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ANALYSIS

Integrating environmental and economic performance to assess modern silvoarable agroforestry in Europe

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ABSTRACT

The environmental and economic performance of silvoarable agroforestry in Europe is highly variable. Multi-criteria analysis, using the PROMETHEE outranking approach, was used to evaluate the integrated performance of silvoarable agroforestry on hypothetical farms in nineteen landscape test sites in Spain, France, and The Netherlands. The silvoarable scenarios allocated a proportion of the hypothetical farms (10 or 50%) to silvoarable agroforestry at two different tree densities (50 or 113 trees ha^{-1}) on two different qualities of land (best or worst quality land). The status quo (conventional arable farming) was also assessed for comparison. The criteria used in the evaluation (soil erosion, nitrogen leaching, carbon sequestration, landscape biodiversity, and infinite net present value) were assessed at each landscape test site; infinite net present value was assessed under six levels of government support. In France, the analysis showed, assuming equal weighting between environmental and economic performance, that silvoarable agroforestry was preferable to conventional arable farming. The best results were observed when agroforestry was implemented on 50% of the highest quality land on the farm; the effect of tree density (50–113 trees ha^{-1}) was small. By contrast, in Spain and The Netherlands, the consistently greater profitability of conventional arable agriculture relative to the agroforestry alternatives made overall performance of agroforestry systems dependent on the proportion of the farm planted, and the tree density and land quality used.

1. Introduction

European agriculture is facing the need for alternatives. Problems of over-production, low farmer's income, abandonment of rural areas and environmental pollution through intensive production need to be addressed. Recent developments of the European policy decouple direct aids from production and aim at steering the support towards more sustainable use of natural resources (EC, 2005b). Modern silvoarable agroforestry (SAF) is a potential land use solution which, in this context, deserves attention. The system is efficient in terms of resource use (Nair, 1993) and can be both environmentally beneficial and economically profitable. This could improve agricultural sustainability, provide opportunities to diversify farm income, provide new products to the wood industry, and create novel landscapes of high value (Dupraz and Newman, 1997).

Recent investigations have shown that the environmental and economic performance of SAF in Europe is highly variable

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(Graves et al., in press; Palma et al., 2007). This variability results from the interaction of many factors influencing outputs of SAF. For example, different economic and environmental results are obtained in different European regions due to the many combinations of biophysical and management conditions, such as choice of tree and crop species, national legislation, market conditions, and regional policies. Because of this variability, SAF can be more or less profitable than conventional arable systems, the trees can be more or less competitive with crops depending on tree species and biophysical conditions and can have more or less beneficial environmental effects or in some situations they may not be making a significant difference at all.

Within the context of agricultural policy which takes into consideration environmental performance (Pezaros, 2001; van Dijk, 2001), an analysis of land use systems should consider both economic and environmental impacts. However, environmental and economic evaluations are complex and tend to be undertaken separately. Drawing together these separate analyses then becomes impossible, because of the different assumptions and scenarios used. For example, environmental results for agroforestry systems obtained by Udawatta et al. (2002) or Nair and Graetz (2004) cannot be linked to economic results obtained in different contexts (e.g. Thomas, 1991; Thomas and Willis, 1997). Whilst there have been some studies on the economic performance of agroforestry (Willis et al., 1993; Dupraz et al., 1995; Mercer et al., 1998; Requillart et al., 2003; Montambault and Alavalapati, 2005), environmental assessments have been limited to case studies (e.g. Burgess, 1999; Stamps et al., 2002; Montagnini and Nair, 2004; Thevathasan and Gordon, 2004; Klaa et al., 2005) and no integrated assessments have been conducted to date.

Integrated assessments require the comparison of independent indicators (e.g. soil erosion, nitrogen leaching, profitability) with different physical units (e.g. tonnes ha^{-1} $year^{-1}$ of soil loss, kg ha^{-1} $year^{-1}$ of leached nitrogen, € ha^{-1} $year^{-1}$ of profit). One way of integrating the results of such indicators is to monetarise them and compute an overall profitability, which therefore includes the monetary value of the environmental performance. The advantage of this approach lies in the ease of communication of the final result — an integrated profitability. Also, monetarising environmental costs and benefits emphasize their economic significance and importance to individuals and society. However, there are a multitude of difficulties related to monetarisation. Whilst estimates of the economic value of soil loss can be based on impacts on crop yields, or the cost of nitrogen leaching derived from the cost of water purification, the economic value of landscape biodiversity for example is more difficult to assess. Although economic values can be obtained through various methods, such as contingency valuation, the validity of results obtained from such approaches has been questioned (Mitchel and Carson, 1989; Hanley and Spash, 1993; Pethig, 1993). Moreover, an integrated assessment based on monetarisation is not always transparent. Whilst the financial benefit of agricultural production goes to the farmer, environmental benefits (or costs) may be relevant to either the farmer (e.g. soil erosion, which reduces profitability), both the farmer and society (e.g. nitrogen leaching which creates additional fertilizer costs for the farmer and water purification costs for society), or society as a whole (e.g. landscape biodiversity). These distinctions are lost when integrated in a single integrated monetary value.

Hence, multi-criteria decision analysis (MCDA) was used instead to evaluate the relative importance of the selected criteria and reflect their importance in the final result (Belton and Stewart, 2002). Environmental results (Palma et al., 2007) and economic results (Graves et al., in press) obtained previously during the European Union "Silvoarable Agroforestry for Europe" project (Dupraz et al., 2005) were assessed using MCDA to provide an integrated analysis of the environmental and economic benefits of SAF. This paper therefore provides an integrated overview of the impact of SAF systems which were considered to be suitable for Europe (EC, 2005a; Lawson et al., 2005; Reisner et al., in press).

2. Materials and methods

2.1. *Environmental and economic data*

An environmental classification of Europe, derived from a statistical analysis of climatic and topographic data (Metzger et al., 2005), was used to randomly select 19 landscape test sites (LTS) of 4 km×4 km in the dominant environmental classes of Spain, France and The Netherlands (Graves et al., in press; Palma et al., 2007).

The radiation, temperature, rainfall, soil depth and texture data for each LTS were used as inputs in a daily time-step bio-physical model of tree and crop production, based on competition for light and water (Yield-SAFE, van der Werf et al., in press) and implemented in Microsoft Excel© by Burgess et al. (2004) to predict annual tree and crop yields. Scenarios comprised hypothetical farms with SAF at two densities (50 and 113 trees ha^{-1}, 40×5 m and 22×4 m respectively) on 10 and 50% of the total agricultural area, starting with either the best and worst quality land. Current agricultural land use was also modeled to provide a comparison with the status quo.

For each scenario in each LTS, the environmental assessment comprised analysis of soil erosion, nitrate leaching, carbon sequestration and landscape biodiversity (Palma et al., 2007, in press). Erosion was modeled with the revised universal soil loss equation (RUSLE, Renard et al., 1997), where SAF was considered to mimic strip cropping, implemented together with contour farming. Nitrogen leaching was modeled using an equation proposed by Feldwisch et al. (1998), which uses an annual water exchange factor in the soil and the excess nitrogen potentially available for leaching. Annual excess nitrogen was estimated from tree and crop productivity, assuming optimized nitrogen fertilization, taking into account nitrogen contents of crop-tree biomass, of the soil and the nitrogen recovery capacity by crops (van Keulen, 1982). Carbon sequestration was calculated for SAF systems only, based on the Intergovernmental Panel on Climate Change (IPCC, 1996) and Gifford relationships (2000a,b) for tree biomass predicted by the Yield-SAFE model. A broad evaluation of the effects of SAF implementation on landscape biodiversity was conducted, based on the share of habitats available to wildlife in an agricultural landscape, classifying each LTS into “habitat” (e.g. hedgerows, permanent grassland, traditional orchards) and “non-habitat” (the arable matrix). Further details on the environmental assessment can be found in Palma et al. (2007).

Graves et al. (in press) used the Yield-SAFE predicted annual yields of trees and crops and financial data collected through workshops held in each country as inputs for a plot- and farm-scale cost–benefit economic model called “Farm-SAFE” (Graves et al., 2005). Profitability was assessed, in terms of the infinite net present value (iNPV) for six levels of government support of SAF. The first scenario considered no support at all (iNPV_0), the second considered the support received from the European Union’s Common Agricultural Policy (CAP) up to 2004 (iNPV_04) and four schemes considered future options based on CAP reform in the Rural Development Regulation (EC, 2005a) and included use of single farm payments (SFP) (EC, 2004). The first of these considered that SFP would be based on the percentage of crop area in silvoarable system (iNPV_05_1.1), the second assumed SFP for the whole area (iNPV_05_1.2), the third (iNPV_05_2.1) considered SFP as for iNPV_05_1.1, but included additional tree payments as outlined in the Rural Development Regulation, and the fourth (iNPV_05_2.2) considered SFP as for iNPV_05_1.2, but with the additional tree payments. The tree payments were assumed to be equivalent to half of the costs of tree establishment during the first four years of the tree rotation (see Graves et al., in press for details).

In order to harmonize with MCDA terminology, the “assessments” and “land use scenarios” will be called criteria and alternatives, respectively.

In each LTS, the environmental and economic criteria for SAF were modeled for different alternatives of tree density (50 or 113 trees ha^{-1}), land quality (best or worst) and share of farmland converted to SAF (10 or 50%). Tree lines were assumed to be planted along contour lines, which is important for erosion control (Palma et al., 2007). A total of eight alternatives were compared with conventional arable land use (status quo) under six different payment schemes (Fig. 1).

2.2. MCDA outranking procedure

MCDA outranking methods focus on pair-wise comparisons of alternatives where the starting point is a decision matrix describing the performance of the alternatives to be evaluated with respect to identified criteria (Belton and Stewart, 2002). The PROMETHEE II method (Brans and Vincke, 1985; Brans et al., 1986; Brans and Mareschal, 1990, 2002) was used as it enables the complete pre-order of alternatives, facilitating the tracing of the final performance rank.

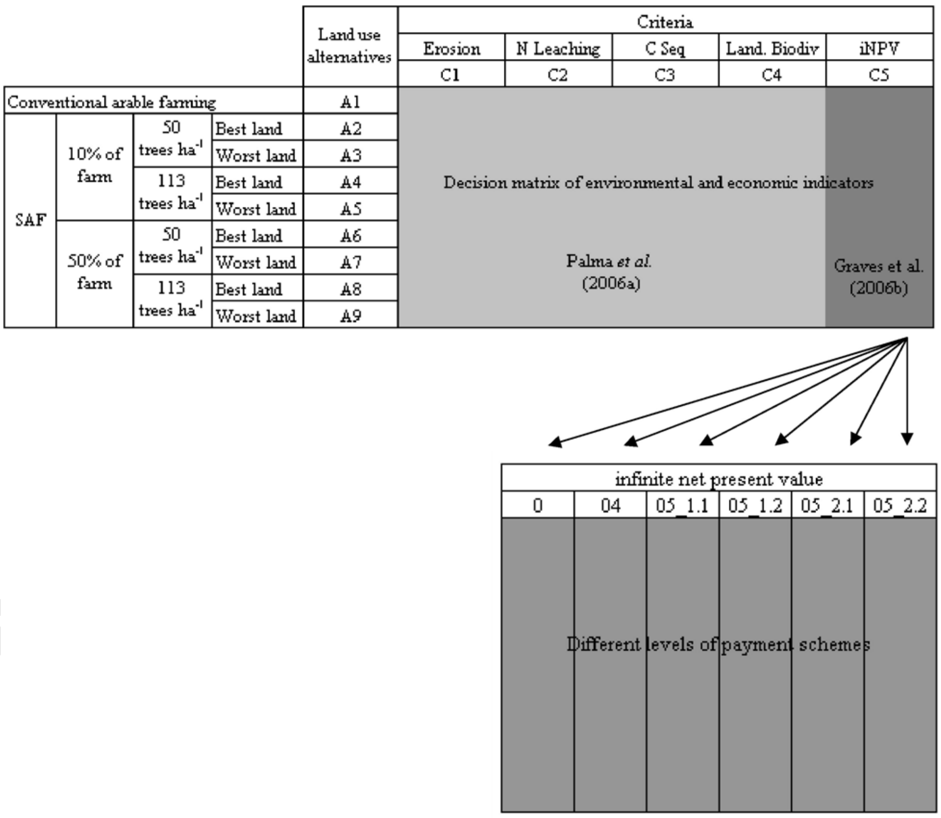

Fig. 1 – Definition of the alternatives and design of the decision matrix with floating levels of payments for the economic criteria. SAF, Silvoarable Agroforestry; C Seq, Carbon Sequestration; Land. Biodiv., Landscape Biodiversity; iNPV, infinite net present value; see codes of the economic scenarios in Section 2.1.

A general characteristic of PROMETHEE and other outranking methods is that all k alternatives are compared in a pair-wise manner, separately for each criterion. To formalize this, let $g_j(L_a)$ and $g_j(L_b)$ be the values of two land use alternatives L_a and L_b for criterion C_j. The difference between the two indicator values is denoted as $d_j(L_a, L_b) = g_j(L_a) - g_j(L_b)$, which measures the extent to which L_a 'outperforms' L_b in criterion C_j. The "preference function" Π_j (L_a, L_b) maps this difference into a preference score which is between 0 and 1 and mirrors how strongly L_a is preferred to L_b in terms of C_j. We opted for the "Type II — quasi-criterion function" with zero as the reference parameter: if L_a outperforms L_b, i.e. if $d_j(L_a, L_b) > 0$, then the preference of L_a to L_b is Π_j $(L_a, L_b) = 1$. If L_a and L_b perform equally or if L_b outperforms L_a, i.e. if $d_j(L_a, L_b) \leq 0$, then the preference of L_a to L_b is Π_j $(L_a, L_b) = 0$.

The preference was calculated separately for each criterion and for all pairs of land use alternatives. These preferences were aggregated over the criteria to obtain a total preference for each pair of land use alternatives. The "total preference" of L_a to L_b was then calculated as the weighted sum of the pair-wise preferences for all criterion indicators ($j = 1$ to 5), the latter associated to a certain weight of importance (w):

$$\Pi(L_a, L_b) = \sum_{j=1}^{5} w_j \Pi_j(L_a, L_b) \quad \text{with} \sum_{j=1}^{5} w_j = 1 \tag{1}$$

In this primary evaluation, the weights (w) were given according to a neutral preference (see Section 2.4 below).

Eq. (1) reflects only two land use alternatives (L_a, L_b). In the case of this evaluation there were nine alternatives. Therefore the total preference was calculated for all pairs of land use alternatives (L_u, L_v) ($u,v = 1$ to 9). The nine alternatives were ranked in two different ways. The first reflects how strongly an alternative L_u dominates all the other alternatives L_v ($v = 1$ to 9) – $\Phi^+(L_u)$ in Eq. (2) — and the second reflects how strongly L_u is dominated by all the other alternatives L_v ($v = 1$ to 9) – $\Phi^-(L_u)$ in Eq. (3).

$$\Phi^+(L_u) = \sum_{v=1}^{9} \Pi(L_u, L_v) \tag{2}$$

$$\Phi^-(L_u) = \sum_{v=1}^{9} \Pi(L_v, L_u) \tag{3}$$

Finally, a performance rank Φ (L_u) considering all alternatives was computed for each alternative:

$$\Phi(L_u) = \Phi^+(L_u) - \Phi^-(L_u) \quad \text{with } u = 1, \ldots, 9 \tag{4}$$

Each land use alternative was then ranked according to the integrated environmental and economic performance of each alternative. The higher the performance rank, the higher the preference of the alternative.

2.3. MCDA design

The MCDA had nine alternatives and five criteria to evaluate (Fig. 1). For each of the nineteen LTS, a decision matrix was built (Fig. 1) with all the criteria assessed by Palma et al. (2007) and Graves et al. (in press). The erosion and nitrogen leaching assessment values were rescaled in order to show reduction of erosion and nitrogen leaching compared to the status quo, so higher values could correspond numerically to "better alternatives".

Graves et al. (in press) modelled six different economic payment schemes under different subsidy levels, therefore six decision matrices were built for each LTS, keeping constant the environmental indicators and varying the economic indicator (Fig. 1). A total of 114 decision matrices were analysed.

In each LTS, the alternatives were ranked with Φ (Eq. 4) and an average of the performance of each alternative was calculated for all 19 LTS. A subgroup average at country level was also calculated as the integrated economic and environmental results varied substantially within each country (Graves et al., in press; Palma et al., 2007).

2.4. Weighting criteria

Different stakeholders can have different objectives depending on their personal values and on their socio-economic circumstances. As we did not wish to adopt the position of a specific stakeholder (e.g. an NGO might rate environmental criteria higher than profitability, whereas farmers might rate profitability higher than environmental criteria), for this MCDA a 'neutral' weight distribution was used. Thus, the environmental and economic criteria were considered as two groups with the same weight (0.5 each). Because the environmental assessment involved four criteria, each one had the value of (0.5/4). In other words, the sum of all the environmental outputs of each land use alternative was tested against its economic performance.

For a better starting point in the evaluation, we also ran the analysis in a "uni-criterion" analysis mode to have an overview of the performance of each alternative under each criterion independently. This was done using Eq. 1, by setting the weight of the selected criterion to 1 and the weights of all other criteria to 0.

3. Results and discussion

3.1. "Uni-criterion" analysis

The performance of each alternative was evaluated for each criterion independently to facilitate understanding of how each alternative was influenced by individual criteria. The results are shown on a per country basis (Fig. 2).

The difference in effect between environmental and economic criteria are clearly visible in Fig. 2. The aggregate result for all 19 LTS shows that alternatives with the best environmental performance were those with the worst economic performance (Fig. 2a) except for iNPV_0. Under this "zero subsidy" assumption, the alternative of implementing SAF with 50 trees ha^{-1} on 10 or 50% of the farm showed similar or slightly higher preference than the status quo alternative. Under all levels of government support however, the status quo alternative was preferable to SAF. But SAF was most preferable when soil erosion, nitrogen leaching, carbon sequestration and landscape biodiversity were evaluated. The best environmental and worst economic evaluations

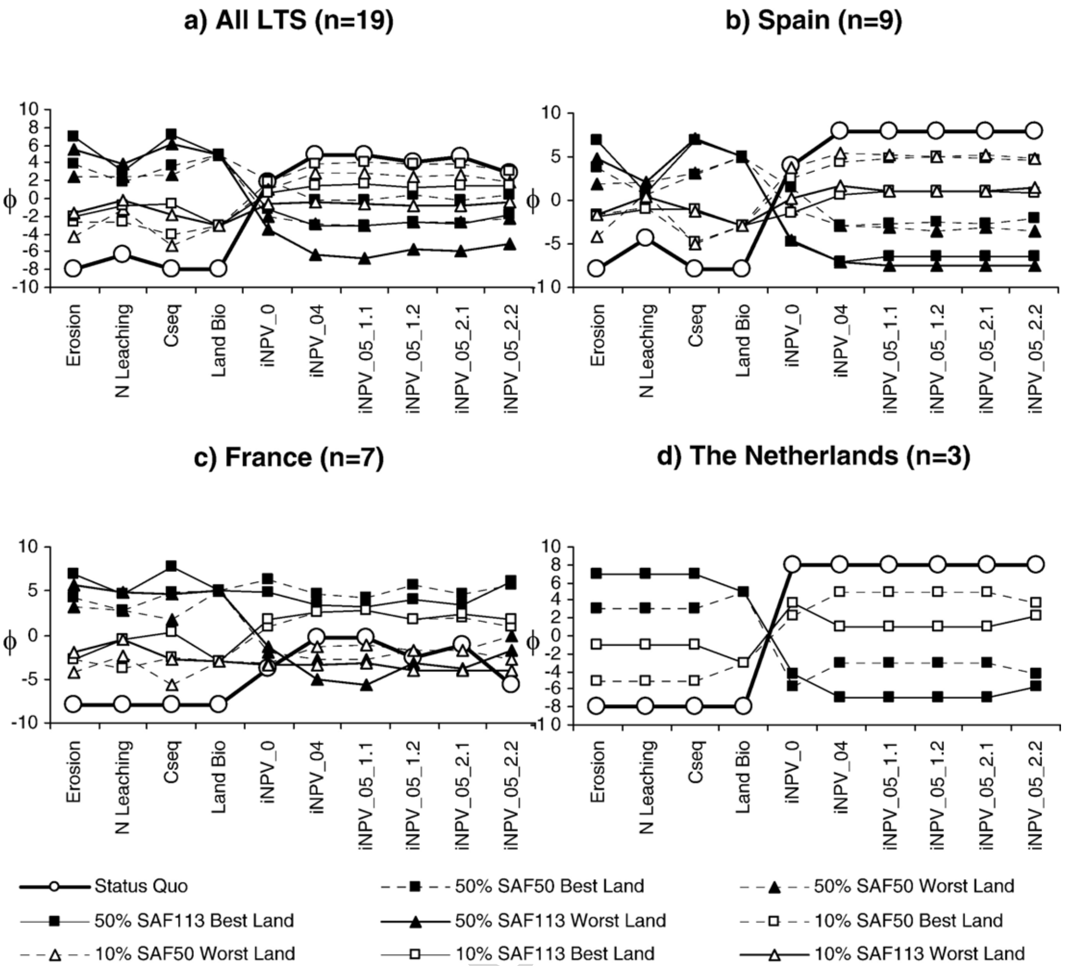

Fig. 2 – Preference rank of each alternative for each criterion for a) all landscape test sites (LTS) and for b) Spain, c) France and d) The Netherlands separately. Note: in The Netherlands there was only one homogeneous landscape and therefore only one quality of land was assessed.

were associated with SAF implemented on 50% of the farm (filled symbols).

These patterns were also observed in Spain and The Netherlands (Fig. 2b,d). In France however, the SAF alternatives on high quality land were preferable to the status quo alternative for all environmental and economic criteria, regardless of the tree density or the proportion of the farm to be converted (Fig. 2c). This was in part due to relatively high financial returns from the timber produced in the SAF systems, since the trees selected were valuable (walnut, *Juglans* hybr.) or produced rapid returns (poplar, *Populus* sp.) (Graves et al., in press).

3.2. *Multicriteria analysis*

In the next step, the environmental and economic criteria were analyzed together. Generally, the performance rank Φ of arable systems was higher for the scenarios assessing subsidy payments than for the "zero subsidy" scenario. The contrary occurred for the SAF alternatives. In other words, the subsidiary schemes distort profitability in favor of arable systems. Nevertheless, the aggregate results for all 19 LTS show that the SAF alternatives were preferable to the status quo alternative under all payment schemes and especially when these are absent (Fig. 3a — iNPV_0). However, this aggregate response for all three countries masks important differences between the countries that occurred as a result of regional variations in biophysical conditions, selected tree and crop species, and market dynamics.

In Spain, government policies on trees and crops favored conventional arable agriculture relative to agroforestry (see Palma et al., 2004; Graves et al., in press) as shown by higher ranking of the status quo under government support (Fig. 3b). Even so, the SAF alternatives with low tree density on 10% of the worst land were preferable to the status quo (Fig. 3b — open symbols and dotted lines). In the absence of government support (Fig. 3b — iNPV_0), SAF alternatives performed considerably better than the conventional arable alternative, especially in cases where 50% of the farms were converted to SAF.

In France, the final evaluation was less affected by differences in the payment schemes and depended more on the financial productivity of the systems (i.e. the income generated by selling wood and crops). In fact, for all payment schemes, SAF alternatives performed better than the status quo. The best results were related to SAF implementation on

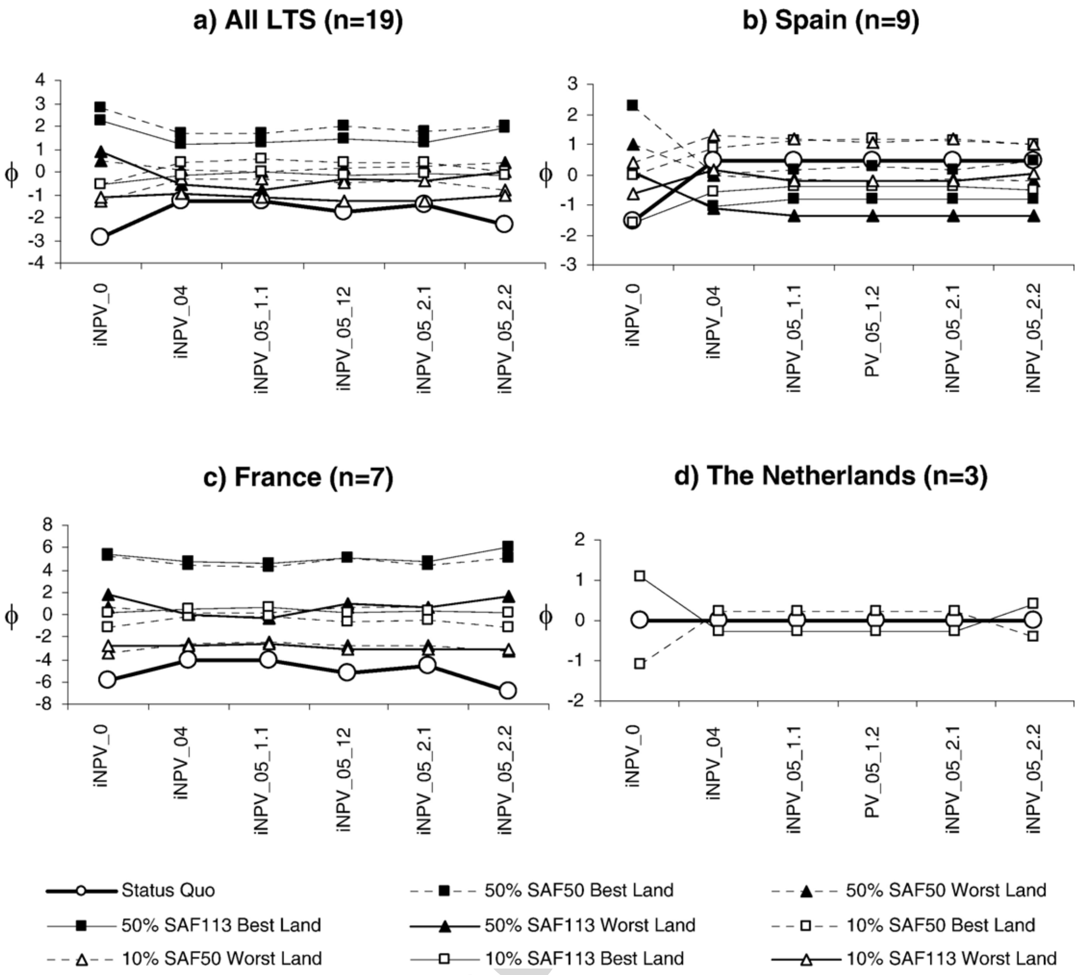

Fig. 3 – Preference rank of alternatives under six economic payment schemes a) for all landscape test sites (LTS) and b) for Spain, c) for France and d) for The Netherlands separately. The weight attribution represents an equilibrium evaluation between environmental and economic criteria (see Section 2.4). Note: in The Netherlands there were only homogeneous landscapes, therefore only one quality of land was assessed.

50% of the high quality land on the farm (Fig. 3c — filled square symbols). The improved performance on high quality land is partly associated with the decision that high value walnut trees would tend to be allocated to such areas, whereas lower value wild cherry (*Prunus avium* L.) and poplar trees were generally allocated to lower quality land (see Graves et al., in press). Even so, on low quality land, wild cherry and poplar agroforestry systems often outranked the status quo arable cropping (Fig. 3c).

In The Netherlands, land quality over the farm was homogenous and only the existing quality land was assessed (termed "best land" in Fig. 3d). The 10% and the 50% SAF alternatives (open and filled square symbols, respectively, in Fig. 3d) overlap, which indicates that the area of implementation does not play a role in the preference between SAF alternatives in this country. The SAF alternatives with 113 trees ha^{-1} either on 10 or 50% of the farm were preferable to the other alternatives (Fig. 3d — continuous line), and the relative ranking of arable cropping was not influenced by the subsidies as observed in Spain or France. However, the subsidies negatively affected the performance of SAF at higher tree densities (113 trees ha^{-1}) (Fig. 3d — continuous lines), and positively affected lower densities. These effects are partially recovered with tree payments in scenario iNPV_05_2.2.

4. Conclusions and further research

In MCDA the ecological impacts of SAF are usually assessed using qualitative approaches (e.g. Sipos, 2005). The evaluation presented here used quantitative environmental and economic outputs as criteria in an integrated assessment of SAF in comparison with conventional agriculture.

The land use alternatives were designed to reflect the questions that typically concern farmers, for example, regarding the proportion of the farm which should be allocated to the new land use, the land quality which should be used, and the tree density of planting. The evaluation showed that in order to achieve environmental benefits, larger portions of the better quality land should be converted; whereas tree density was a less important consideration (Fig. 2). Economic performance was improved if high value trees were established on the highest quality farmland, rather than lower value trees on low quality land. Thus, the main conclusion to be drawn from these findings is that new SAF systems in Europe will generally have the greatest benefit if quality timber trees are established over substantial areas of the farm and on the best quality land.

The intention of the alternative payment schemes was to derive recommendations on the type and level of public subsidies in order to render SAF (economically) competitive with conventional arable cropping and therefore to "harvest" its environmental benefits. However, the CAP reforms implemented in 2005 were predicted to have only a minimal effect on the profitability of SAF relative to the pre-reform situation (iNPV_04). For example, whether the SFP was available for the cropped area (iNPV_05_1.1) or the whole area (iNPV_05_1.2) made a difference only in France, where the competitiveness of SAF relative to the status quo was increased. Additional payments for tree establishment (schemes iNPV_05_2.1 and iNPV_05_2.2) made minimal difference in Spain and The Netherlands, while in France this effect was more important to the relative performance of the SAF and arable cropping alternatives.

During the coordination of the assessment to retrieve economic and environmental results, problems and challenges arose due to country, even regional particularities. These ranged from different national soil classifications systems and availability of data, to regional particularities that imposed restrictions to certain subsidies. However these challenges were overcome by interaction in local workshops with local experts which accessed relatively easily the local information needed.

The cross country results showed that very different results can be obtained depending either on policy or on biophysical conditions. Policy and biophysical conditions are somewhat related (e.g. regarding nitrogen usage, in The Netherlands there is strong legislation, while in Spain, there is no such problem in rainfed systems due to water scarcity, the vector of leaching). A general EU recommendation can be drawn by averaging results from all countries, recognizing that SAF systems are beneficial "in general". This leads to the recommendation that at the European level, laws and regulations should at least not hamper the introduction of SAF. However, SAF should be implemented with a specific purpose — economic and/or environmental. This purpose needs to be specified at the country/regional level based on local conditions, environmental hazards and socio-economic requirements.

Further research is needed to refine the approach presented here. For example, the financial analysis assumed that farmers would have access to capital that could be used for establishing SAF plantations, but lack of such access could limit SAF, especially in poorer areas of Europe. The analysis was also limited to a comparison of SAF and arable systems and a fuller analysis should include forestry systems. The evaluation could also be improved by weighting the selected criteria to reflect their relative importance in different regions or for different stakeholders (Section 2.4). For example, in Spain, farmers may feel soil erosion control is more important than nitrogen leaching, which is rarely problematic in non-irrigated Mediterranean conditions, whereas in The Netherlands, nitrogen leaching is a major problem, but soil erosion is less of an issue (Palma et al., 2007).

This paper provides an approach for integrated environmental and economic analysis of SAF systems. Such results could be used to support policy development for SAF as a new land use alternative for farmers (Lawson et al., 2004; EC, 2005a; Lawson et al., 2005). Beyond the application to agroforestry, the framework can be expanded to testing other alternative land use systems such as new crops, agricultural energy fuel production, etc. Environmental and economic consequences of alternative land use systems can be truly compared if the investigations are co-ordinated, relate to the same (test) regions and adopt the same scenarios. If ecological integrity is seen as fundamental to economic and social well-being (Kay and Schneider, 1994), assessments of future land use options need to integrate both economic and environmental criteria.

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